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LAND USE IN LCA

Assessment of land use impacts on soil ecological functions: development of spatially differentiated characterization factors within a Canadian context

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Abstract
Purpose Among other regional impact categories in LCA, land use still lacks a suitable assessment method regarding the least developed "soil ecological quality" impact pathway. The goals of this study are to scope the framework addressing soil ecological functions and to improve the development of regionalized characterization factors (CFs). A spatially explicit approach was developed and illustrated for the Canadian context using three different regional scales and for which the extent of spatial variability was assessed.
Materials and methods A model framework based on the multifunctional character of soil and the ecosystem services defined by the Millennium Ecosystem Assessment is suggested. This framework includes land use impacts on soil ecological quality evaluated regarding the change in soil capacity to fulfill a range of soil ecological functions. Four impact indicators, namely erosion resistance, groundwater recharge, mechanical, and physicochemical filtration, proposed by the functional method of Baitz (2002), were used to assess three major degraded regulating services: erosion regulation, freshwater regulation, and water purification. Spatially differentiated CFs were calculated based on the principles proposed by the UNEP/SETAC Life Cycle Initiative for two Canadian spatial models (15 ecozones, 193 ecoregions) along with a non-spatial one (one generic). Seven representative land use types were tested.
Results and discussion Using the ecozone-based scale, an overall result comparison between the non-spatial and spatial models indicates significant differences between ranges across land use types and results up to four times larger than what the generic scale can capture. This highlights the importance of introducing a regionalized assessment. When considering the impacts from a specific land use type, such as urban land use, generic CFs fail to adequately represent spatial CFs because they tend to be highly dependent on the biogeographical conditions of the location. When comparing all three resolution scales, CF results calculated using the ecoregions spatial scale generally show a larger spread across each land use type. Interesting variations and extreme scenarios are revealed which could not be observed using a coarser scale-based model such as the ecozone resolution scheme.
Conclusions This work demonstrates the accomplishment of developing spatially differentiated CFs addressing impacts of different land use types on soil ecological functions. For a large territorial area spreading over many biomes, such as Canada, accounting for ecological unit boundaries proves to be necessary since the generic scale is not sufficiently representative. An evaluation of the extent of spatial differentiation emphasized the influence on the variability of regionalized CFs.

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1 Introduction

Soil is not only a support for human settlements and several economic activities but also a nonrenewable resource due to its extremely slow renewal rate (Pimentel et al. 1987). In order to ensure its availability for future generations and given the relevance of its impacts on ecosystem quality, the sustainable use of land requires informed decisions based on a meaningful impact assessment. Despite many attempts made to account for its environmental impacts in Life Cycle Assessment (LCA) studies, a lack of global consensus among impact assessment methods remains. Indeed, the various existing methodologies accounting for land use impacts are either too restrictive on their spatial coverage (Schmidt 2008) or too selective on the impact pathway. They are generally limited to the European continent and fail to address particular ecosystems related to other countries. Thus, the need for integrating and harmonizing impact indicators is extensively discussed in the literature (Müller-Wenk 1998; Lindeijer et al. 2002; Milà i Canals et al. 2007a).

On one hand, methodological gaps are mainly related to the number of modeled impact pathways. So far, several studies have only focused on the assessment of land use impacts considering effects on biodiversity (Müller-Wenk 1998; Lindeijer 2000; Weidema and Lindeijer 2001; Koellner and Scholz 2007) and biomass production (Cowell and Clift 2000; Lindeijer 2000; Mattsson et al. 2000; Weidema and Lindeijer 2001; Milà i Canals et al. 2007a). However, apart from biodiversity loss (Meyer and Turner 1992), land use change is highlighted as a primary source of soil degradation (Tolba et al. 1992) and a key human-induced effect on natural ecosystem functioning (Lambin et al. 2001). Its related impacts affect not only the ecological quality of soil through a modification of its properties and land cover but also undermine its productivity and other capacities to support human needs by performing many ecological functions (Vitousek et al. 1997; Foley et al. 2005). As suggested by Koellner and Scholz (2008), a harmonized framework should complement existing impact methods and integrate other instrumental aspects for a broader land impact assessment such as ecosystem services. The latter are broad services supported by soil ecological functions, describing the least explored "ecological soil quality" impact pathway and which falls within the framework suggested by the resources and land use UNEP/SETAC Life Cycle Initiative task force (Milà i Canals et al. 2007a). In spite of its significant importance, little attention has been given to this specific pathway in Life Cycle Impact Assessment (LCIA) (Lindeijer et al. 2002), and only a few approaches have been published on the topic. One study proposed the use of soil organic matter as a single soil quality indicator. However, this seems appropriate for agricultural and forestry systems only and insufficient for accounting all aspects of soil ecological functions and other major impacts related to soil degradation such as erosion (Milà i Canals et al. 2007b). A second method (Baitz 2002) suggests a series of indicators based on a functional approach. It addresses effects induced by several anthropogenic land uses having different land covers and sealed surface intensities on a range of more detailed soil ecological functions.

On the other hand, impacts from land use are highly influenced by the condition of the location and depend on many biogeographical factors regarding landscape, climate, and vegetation patterns as well as a range of soil properties (Milà i Canals et al. 2007a). There is a growing recognition of the necessity to develop characterization factors (CFs) accounting for spatial differentiation for such non-global impact categories (Potting and Hauschild 2006). Although site-generic data are often suggested, they can lead to results that do not reflect the impacts accordingly, resulting in uncertainty linked to spatial variability (Hertwich et al. 2002; Sedlbauer et al. 2007). Hence, it is appropriate to account for a regionalized assessment method using an appropriate scale-based resolution scheme. Two types of spatial scales are generally adopted in LCIA: (1) political boundaries (Bare et al. 2003) and (2) biogeographical unit boundaries (Toffoletto et al. 2007; Humbert et al. 2009). However, neither one has been accounted for when developing land use impact CFs within LCIA.

The goals of this study are (1) to scope the framework addressing land use impacts on soil ecological functions; (2) to develop a spatial approach assessing land use impacts on a regional scale; and (3) to illustrate the application of this method by (1) developing spatially differentiated CFs for Canada at different regional scales and (2) evaluating the extent of their spatial variability.

2 Materials and methods

2.1 Framework for addressing land use impacts on ecosystem services potential

Given the multifunctional character of soil (Nortcliff 2002), the evaluation of its ecological quality goes beyond its productive capacity. Rather, it is carried out based on the assessment of the performance of soil to fulfill a range of intended ecological functions. Soil ecological quality is a

fundamental concept in soil science used to bridge the use function and the protection aspects of soil management (Tóth et al. 2007).

The framework proposed in this paper and linked to the work of the UNEP/SETAC Life Cycle Initiative working group on land use LCIA (LULCIA) (LULCIA 2008) is structured according to the classification developed within the Millennium Ecosystem Assessment (MEA) (MEA 2005). Table 1 shows a cross-tabulation between a range of ecological functions generally fulfilled by the soil and the more general ecosystem services to which they contribute at a larger scale. A comprehensive assessment method should enable the evaluation of all these services.

This paper focuses on three major ecosystem services reported by the MEA to be significantly modified due to anthropogenic activities and consequently degraded:

1. Erosion regulation potential (ERP)
2. Freshwater regulation potential (FWRP)
3. Water purification potential (WPP)

The remaining ecosystem services are addressed by other developments in progress within the LULCIA work project of the UNEP/SETAC Life Cycle Initiative group (LULCIA 2008), namely: (4) the biotic production captures the capacity of the ecosystem to produce biomass and refers to the supporting service of biotic primary production (BPP) (Brandão et al., personal communication), (5) carbon sequestration potential models the amount of carbon uptake from the air and refers to the climate regulation service (CSP) (Müller-Wenk and Brandão 2010). In addition, biodiversity damage potential is assessed through the changes in species richness and distribution due to land use activities (De Baan and Koellner, personal communication).

2.2 Characterization model

Two types of environmental interventions are traditionally considered in life cycle inventories (LCI): land occupation and land transformation. Both are distinguished per land use type. Entries regarding the land use type, the area used and/or transformed (square meters) and the duration of use (year) are generally specified as elementary flows (Lindeijer et al. 2002). Impact assessment for both types of interventions is performed based on the framework for land use LCIA suggested by Milà i Canals et al. (2007a). The magnitude of land use impacts is calculated by the area between the curves shown in Fig. 1 and expresses the change in soil ecological quality over time in respect to a reference state. Two simplifying assumptions were made. First, linear shapes were considered to describe the evolution of soil ecological quality over time. Second, the decrease in quality during the occupation phase was shown to be comparatively small to the quality drop during the transformation phase and was thus assumed to be negligible (Lindeijer et al. 2002). A specific quality curve can be developed for each impact indicator and each land use type. In a spatially explicit assessment, information on the ecosystem type supporting a given activity (soil parameters and reference state) must be collected.

Land occupation impacts consist of a postponement of the relaxation period preventing the soil ecological quality (Q_{use}) to evolve back through a spontaneous or assisted regeneration. Its magnitude is coarsely approximated by the area of a parallelogram shape. Land transformation impacts represent a change in land quality during the relaxation period and its magnitude is coarsely approximated by a triangular area. Impact scores can be expressed as per the general Eq. 1 and further specified for land occupation and land transformation as per Eqs. 2 and 3, respectively:

$$\text{Impact} = \text{Inventory flow} \times \text{CF} = A \int \Delta Q(t)\mathrm{d}t \tag{1}$$

$$I_{\mathrm{occ}} = A \times t_{\mathrm{occ}} \times \mathrm{CF}_{\mathrm{occ}}, \text{where } \mathrm{CF}_{\mathrm{occ}} = Q_{\mathrm{relax}} - Q_{\mathrm{use}} \tag{2}$$

$$I_{\mathrm{trans}} = A \times \mathrm{CF}_{\mathrm{trans}}, \text{where } \mathrm{CF}_{\mathrm{trans}} = (Q_{\mathrm{relax}} - Q_{\mathrm{use}}) \times \frac{1}{2} \times t_{\mathrm{relax}} \tag{3}$$

I_{occ} and I_{trans} are, respectively, the impact scores of land occupation and transformation. The inventory flow is $A \times t_{\mathrm{occ}}$ for land occupation and A for land transformation. The parameter A (square meters) is the occupied or transformed area; t_{occ} (year) is the duration of occupation stage (t_2 to t_3 from Fig. 1); t_{relax} (year) is the relaxation time needed for the land to recover (t_3 to t_4 from Fig. 1). The terms Q_{use} and Q_{relax} refers, respectively, to the soil ecological quality at the use phase and the relaxation time stage. They are measured with regards to each soil ecological function. The assessment of permanent quality change, also identified as irreversible impacts (Weidema and Lindeijer 2001), can be measured as the difference between Q_{relax} and the pre-transformation state, also defined as the initial one (Q_{initial}). However, Milà i Canals et al. (2007a) suggest interpreting these permanent changes on a case-specific basis.

CF represents a difference in soil ecological quality where a positive value expresses a reduction of functional capacity and a negative value is a credit to the land use activity on the performance of the soil function. Considering that the area used or transformed (A) and the occupation time (t_{occ}) are inventory data for defining the elementary flows, the development of CFs is the main focus of this paper.

Table 1 Comparison of soil ecological functions with their corresponding ecosystem services according to MEA and the ones chosen by the UNEP/SETAC LCI working group LULCIA

Soil ecological functions (Seybold et al. 1998; Nortcliff 2002)	Ecosystem services (MEA 2005)	Ecosystem services (UNEP/SETAC Life Cycle Initiative LULCIA)
Physical stability and support	Erosion regulation	Erosion Regulation Potential (ERP): Capacity of ecosystems to stabilize soil and to prevent sediment accumulation downstream
Regulate and partition water flow and storage	Water regulation and cycling	Fresh Water Regulation Potential (FWRP): (a) Capacity of ecosystems to regulate peak flow and base flow of surface water; (b) Capacity of ecosystems to recharge ground
Filter, buffer, degrade, immobilize organic and inorganic substances	Water purification and waste treatment	Water Purification Potential (WPP): Physicochemical and mechanical capacity of ecosystems to clean a polluted suspension
Soil fertility and biotic production	Primary production	Biotic Production Potential (BPP): Capacity of ecosystems to produce biomass
Nutrient stock and carbon sequestration	Climate regulation	Carbon Sequestration Potential (CSP): Capacity of ecosystems to uptake carbon from air

2.3 Calculation procedure for a regionalized approach

Measuring land quality parameters (Q) requires an explicit relationship between soil attributes and soil functions which may be constructed based on several observations, samplings, and monitoring data from different regions in order to take into account fluctuations and parameter heterogeneity. However, developing such relationships can be a monumental task (Doran and Parkin 1994), especially for a specific site-dependent approach. To overcome this limitation, we adopted the modeling approach proposed by (Baitz 2002) and further developed into a calculation tool model, LANCA® (LANd use indicator value CAlculation) (Beck et al. 2010), to assess the influence of different land use activities on soil ecological functions. The LANCA® method framework was originally used to calculate indicator values for specific cases of land use that can be used within LCA software and databases for unit processes and cradle-to-gate datasets. Among the range of eight functional indicators suggested in Baitz's method (Baitz 2002), four potential impact indicators[1] were selected to describe the chosen three ecosystem services defined in the scope of land use framework. Additional information on the LANCA® calculation tool model is provided in Online Resource 1:

1. Erosion resistance: measured in tons of soil eroded per hectare per year, relates to ERP and represents the performance of a terrestrial ecosystem to resist to its soil loss through erosion. Erosion rate is mainly influenced by the vegetation cover, enhancing soil retention, as well as the type of soil texture, humus content, gravel content, slope, and the land use type. The calculation is based on the universal soil loss equation.
2. Physicochemical filtration: relates to WPP by measuring the ability of soil to act as a sorption matrix and adsorb dissolved substances. This parameter is mainly dependent on the cation exchange capacity (CEC) expressed in centimole of cation fixed per kilogram of soil and is influenced by soil alkalinity (pH), soil texture, and soil management.
3. Mechanical filtration: relates to WPP as well and represents the capacity of soil to mechanically clarify a suspension ensuring groundwater protection. Its ability to filter pollutant fixed to the soil is expressed as the infiltration in the soil profile and more specifically, as the rate of water passing in a given time unit. The latter heavily depends on soil texture, its porosity, and the depth to groundwater.
4. Groundwater recharge: relates to FWRP and measures the ability of soil to recharge groundwater in order to regulate peak flow through the magnitude of runoff and

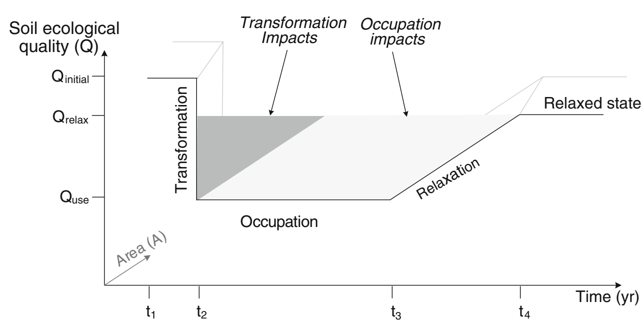

Fig. 1 Soil ecological quality curve indicating the corresponding transformation impact (*dark gray*) and occupation impact (*light gray*); adapted from Lindeijer et al. 2002; Milà i Canals et al. 2007a

[1] The four remaining impact indicators from the method proposed by Baitz (2002) are disregarded in this paper because they either have already been assessed in other existing methods or simply do not fall within the scope of the framework defined for land use impacts on ecosystem services potential (section 2.1): (1) biotic production, (2) pollution control function (particulate and acoustic), (3) species diversity and ecosystem formation, and (4) other functions and potentials of natural areas (landscape quality, recreation esthetic functions, etc.)

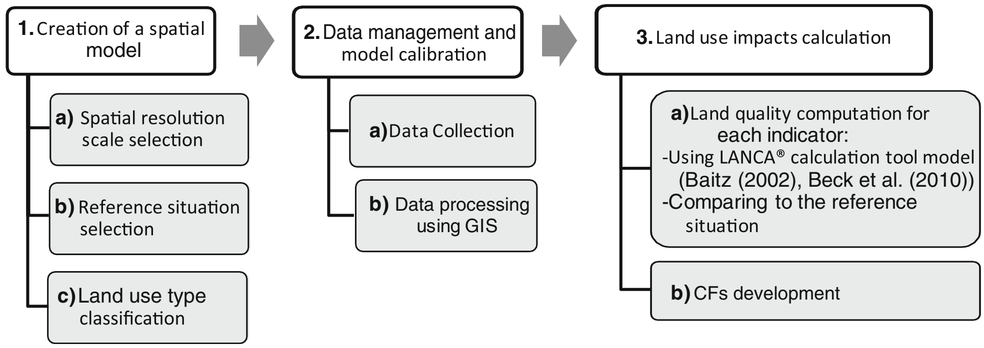

Fig. 2 Calculation procedure for the proposed land use impact assessment regionalized approach

aquifer recharge. Based on the vegetation cover type, climatic, hydrological, and topographical conditions as well as the particle size and the soil texture, the groundwater recharge rate is measured in millimeters of water recharged during a year and modeled using water balance combining precipitation rate, evapotranspiration rate, field capacity, and runoff coefficient.

This method has been chosen because it considers the multifunctional aspects of a soil resource and does not only regard its productive capacity, as opposed to other suggested LCIA land use methods. Moreover, this operational method is reported to be heavily dependent on GIS and data availability (Lindeijer et al. 2002) which makes it interesting for addressing spatial differentiation.

Three main steps are proposed in Fig. 2 to develop an operational regionalized characterization approach for land use impact assessment. The Canadian context is used to illustrate this method.

2.3.1 Creation of a spatial model

Spatial resolution scale selection Creating spatial models begins with defining assessment boundaries and dividing the given region, for instance the country, into different units. Political boundaries delineation such as provinces barely reflects the dynamic nature of ecosystems, especially for countries with a large territorial area such as Canada. Biogeographical unit boundaries are deemed more appropriate because they provide significant information regarding the integrity of natural resources, their management, and environmental analysis. In order to ensure a relevant assessment, two spatially explicit models, based on a commonly used national standardized reference for Canadian ecosystems (Ecological Stratification Working 1995), were created (Fig. 3) based on:

- Fifteen terrestrial ecozones: ecological units describing broad mosaics of ecosystem types, fauna, flora, and geological characteristic
- One hundred ninety-three ecoregions: subdivisions of ecozones characterized by distinctive ecological factors, landscape, macro- or mesoclimate, and plant distribution at a regional scale

Representing a different hierarchical classification level, the ecozone and ecoregion boundaries identify specific ecological regions distinguished by their environmental parameters, biogeographic, and landform characteristics (Statistics Canada 2006). Thus, such boundaries are deemed well adapted for the spatial differentiation of soil ecological functions in Canada. In addition, a generic non-spatial model, considering Canada as one single ecological unit, was also developed for comparative purposes.

Reference situation selection Quantifying land occupation and/or transformation impacts caused by land use activities is measured in relation to a land quality difference between two states (see Fig. 1) and thus requires a reference situation against which the actual state is compared to.

The potential natural vegetation (PNV) state was chosen as the reference situation to present the relaxed state (Q_{relax}) following a land occupation activity. The latter describes the vegetation that would develop if all human influences would stop at once (Westhoff and Van der Maarel 1973). This option is considered the most adequate in the case of attributional LCA and tends to overcome bias in the results when ascribing occupation and transformation impacts (Lindeijer et al. 2002; Milà i Canals et al. 2007a). However, depending on the purpose of the study, the alternative use system is suggested as the reference situation in a consequential modeling (Milà i Canals et al. 2007a).

A PNV type was identified for each ecological unit of the Canadian study area using the output map results provided by BIOME3 model (Haxeltine and Prentice 1996) and a finer spatially resolved one, BIOME4 (Kaplan et al. 2003). The latter are equilibrium terrestrial biosphere models built to simulate the global vegetation distribution of major PNV types. Having different spatial reference and projection systems, a geographic information system (GIS), ArcGIS 9.3 (ESRI 2010), was used to georeference each map image by scaling and aligning the geographic data to match the coordination points of Canada's shapefile. The georeferencing step was established to fit the correct map

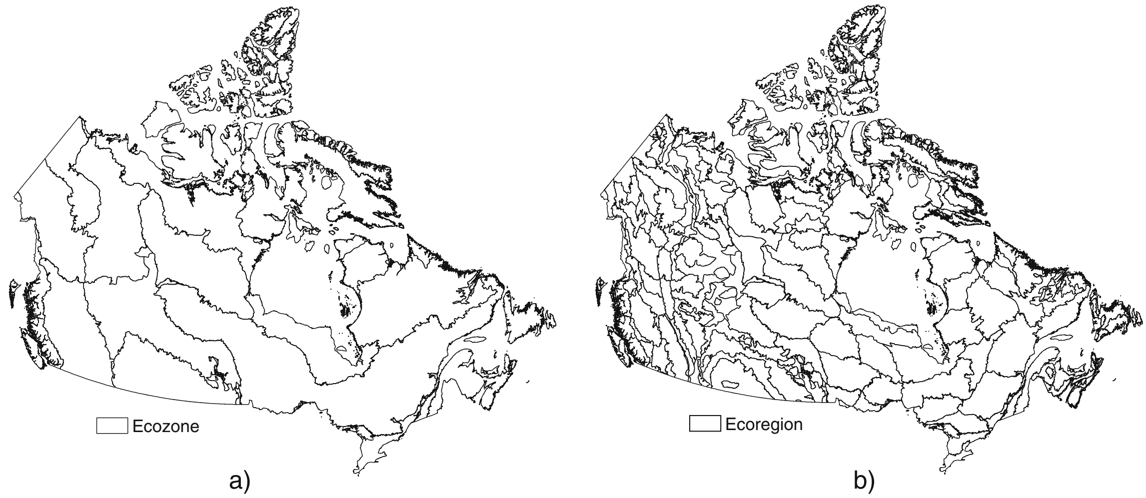

Fig. 3 Graphical representation of the distribution of Canada's ecological units, based on two spatial resolution scales: **a** 15 ecozones and **b** 193 ecoregions

coordinates and performed using a first order polynomial transformation with a root mean square error of less than one pixel.

Land use type classification In order to harmonize their classification, the calculation model land use categories and nomenclature were adapted based on the CORINE land cover classification (Bossard et al. 2000). As indicated in Table 2, a set of seven representative land use types was selected. Such a general classification may be structured into different levels depending on the degree of land use activity (Koellner and Scholz 2008). The set of representative land use types was used for the calculation considering their different influence on soil ecological functions behavior.

2.3.2 Data management and model calibration

To compute soil ecological function results for each land use type and for all four impact indicators, the LANCA® model, based on Baitz's approach (Baitz 2002), requires input parameters consisting of soil properties, landscape, and climatic conditions that are further detailed in the model documentation (Beck et al. 2010). They were collected using several international accessible databases as reported in Table 3. The collected data were managed and processed using ArcGIS 9.3 by overlaying all environmental and landscape parameters and then intersecting them with each resolution scale. For each input parameter, queries were created allowing data analysis and statistical evaluation of spatial information. Therefore, spatially resolved mean values were calculated for all specific ecological units of both spatial Canadian resolution scheme models (ecozones and ecoregions). Mean values for the non-spatially resolved version of Canada used national average parameters, weighted with regards to the ecozone surface area.

Table 2 Selected land use types from CORINE land cover classification

Level 1	Level 2
Artificial	Urban
	Artificial non-agriculture vegetated area
Agriculture	Permanent and annual crops
	Pastures
Forest	Forest
	Grassland
	Shrubland
Wetlands	Excluded

2.3.3 Soil ecological quality computation and development of characterization factors

Using the LANCA® calculation tool model and the appropriate input data, absolute soil ecological quality parameters were calculated for the seven land use types and for each ecological unit of the three resolution scales (one Canada generic, 15 ecozones, and 193 ecoregions). Soil ecological quality parameters (Q_{use} and Q_{relax} from Fig. 1) refer to the quality at the use time stage and the PNV, being selected as the reference state. Hence, CFs were developed for each ecological unit and for all four impact indicators as per Eqs. 2 and 3.[2] Additional information is

[2] The approach adopted in the LANCA model to reconstruct the land quality curve and calculate occupation and transformation impacts is slightly different than the one presented in this paper.

Table 3 Spatially resolved input parameters for the three scale levels

Input parameter		Data range value			Description and source
		Canada generic	Ecozone	Ecoregion	
Soil properties	Soil texture	Loam	All	All	Harmonized Soil Database data sets[a], Canada Ecoatlas[b]
	Organic matter content (%)	8.57	1.85 to 59.03	1.12 to 70.52	SOM can range from 48 to 58% C (Nelson and Sommers 1996). SOM was calculated based on an approximate factor of 1.8 times SOC. The latter was calculated using the Harmonized Soil Database datasets[a]
	Gravel content (%)	11.69	1.12 to 18.27	1.08 to 26.0	Harmonized Soil Database datasets[a]
	CEC ($cmol_c$/kg)	19.97	8.99 to 99.18	5.00 to 120.97	Harmonized Soil Database datasets[a]
	pH	5.89	4.77 to 7.18	4.50 to 11.49	Harmonized Soil Database datasets[a]
Landscape and climatic conditions	Depth to groundwater (m)	Fixed value to 3 m[c]			Since water table levels are highly dynamic and fluctuate over time and seasons, depth to groundwater was considered constant in the model
	Annual precipitation rate (mm/year)	473.84	184.40 to 1914. 90	149.40 to 2202.60	Terrestrial Ecoregions Base Global datasets[d], Canada Ecoatlas[b]
	Annual evapo-transpiration rate (mm/year)	234.02	77.76 to 523.69	25.20 to 580.95	Terrestrial Ecoregions Base Global datasets[d], Canada Ecoatlas[b]
	Slope (°)	1	0 to 25	0 to 25	HYDRO1k Elevation Derivative Database[e]

[a] (FAO et al. 2008)

[b] (Marshall et al. 1999)

[c] Average value (Stone and Myslik 2007)

[d] (Olson et al. 2001)

[e] (US Geological Survey and Earth Resources Observation and Science (EROS) 2009)

provided in Online Resource 1 and the model documentation (Beck et al. 2010).

To facilitate the interpretation and better understand the spatial variability between each resolution scheme, resulting CFs are reported in statistical terms using box plots, each of them grouped by land use type and ecological unit. Supplementary statistical analyses and measures of variability were also performed. Furthermore, a multi-way multidimensional analysis of variance (multi-way MANOVA) of the LANCA® model was conducted (see Online Resource 1), allowing us to partition each result and identify the input parameters that significantly affected the output results (soil ecological quality parameters for each impact indicator).

3 Results and discussion

Only CF results for land occupation are presented in this paper and discussed together with recommendations for land transformation. Detailed tables of CFs for each spatial Canadian model (ecozone and ecoregion-based resolution scale) and the non-spatial model (Canada generic) for all four land use soil ecological functions impact indicators are provided in Online Resource 2.

3.1 Spatial variability analysis of characterization factors

3.1.1 Comparison between the Canadian spatial and non-spatial models

Two types of differentiation were noted to be crucial when addressing spatial variability of land use impacts: land use types and ecological classification systems. The first type of spatial variability aims to compare the impact magnitude from different land use types within the same ecological unit, while the second type considers the impact induced from the same type of activity among a range of ecological units differing by their properties and vulnerabilities.

Figure 4 represents box plots for groundwater recharge capacity CFs developed for the ecozone resolution scale and compared to the generic ones. The box plots illustrate the statistical distributions for both the non-spatial (Canada generic, first box-plot on the left-hand side) and the spatial

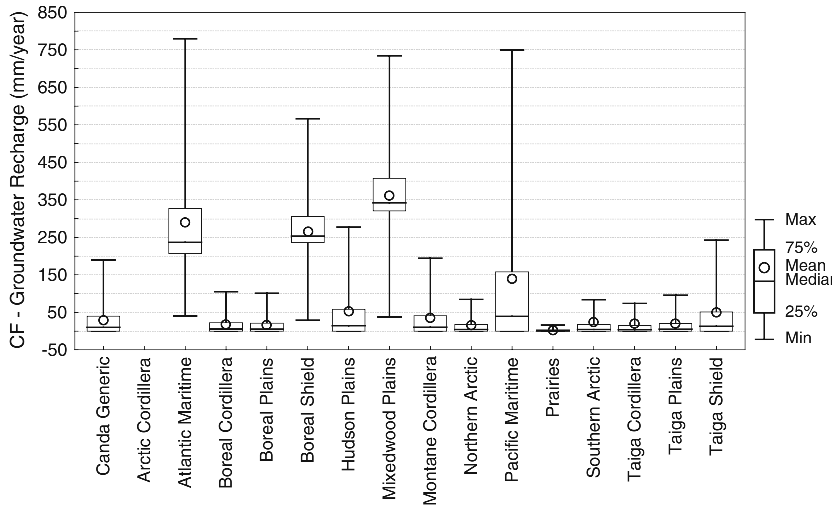

Fig. 4 Overall spatial variability of groundwater recharge CFs results across Canadian ecozones compared to the generic ones

model (Canadian ecozones), indicating the maximum and the minimum CFs, the interquartile range (25th percentile and 75th percentile), the median, and the mean value. For simplicity, results for the remaining three impact indicators are presented in Online Resource 3.

An overall comparison between results from the non-spatial to spatial models shows different distributions for each ecozone, indicating important variability of CFs among different land use types within different ecological units. The observed range for the Canadian generic CFs is much smaller (189 mm/year) than the ones obtained for each ecozone separately, such as the Boreal Shield (537 mm/year), the Mixedwood Plains (696 mm/year), and the Hudson Plains (277 mm/year). More particularly, the case of the Atlantic and the Pacific Maritime shows a measure of variability up to 4 times larger than what the non-spatial Canadian model can capture. Located, respectively, in the southeastern and southwestern coastal parts of Canada, such ecological units are characterized by higher precipitation and evapotranspiration rates than the single average estimates for Canada. Furthermore, median and mean values diverge significantly across quite a few ecological units (up to 34 times higher in median differences for the Mixedwood Plains) and for which the Canadian generic ones are generally situated in the lower part of the results obtained while using a spatial model, indicating skewness in the distribution.

Figure 5 gives an example of the CFs spatial variability across the Canadian ecozones for an urban land use type activity. This type of map shows that, depending on the location, impacts on groundwater recharge capacity can vary up to a factor 50, from 16 mm/year in the Prairie to 779 mm/year in the Atlantic Maritime, as opposed to the generic CF of 189 mm/year.

3.1.2 Spatial variability of characterization factors across ecological units for different resolution scales

The Canadian ecoregion delineation has been analyzed to evaluate the variability of the results at a finer resolution. Figure 6 shows a comparison between three scale levels, ecozone and ecoregion-based spatial resolution, along with

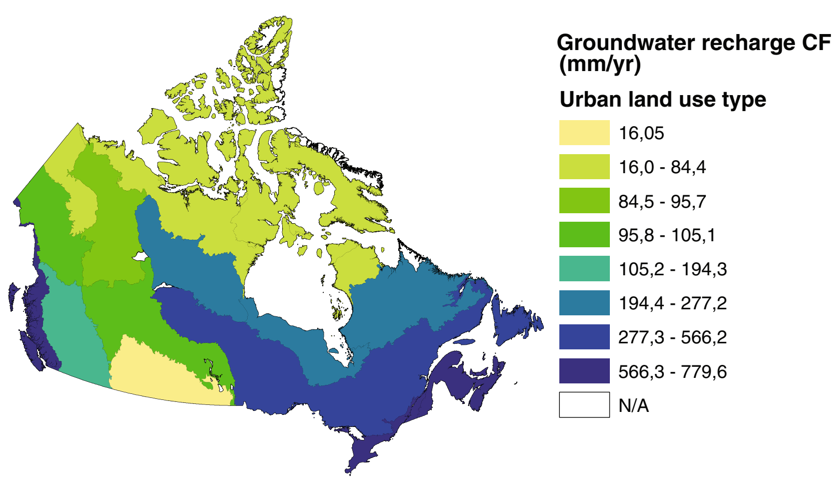

Fig. 5 Magnitude of CFs for groundwater recharge capacity from an urban land use type across Canadian ecozones

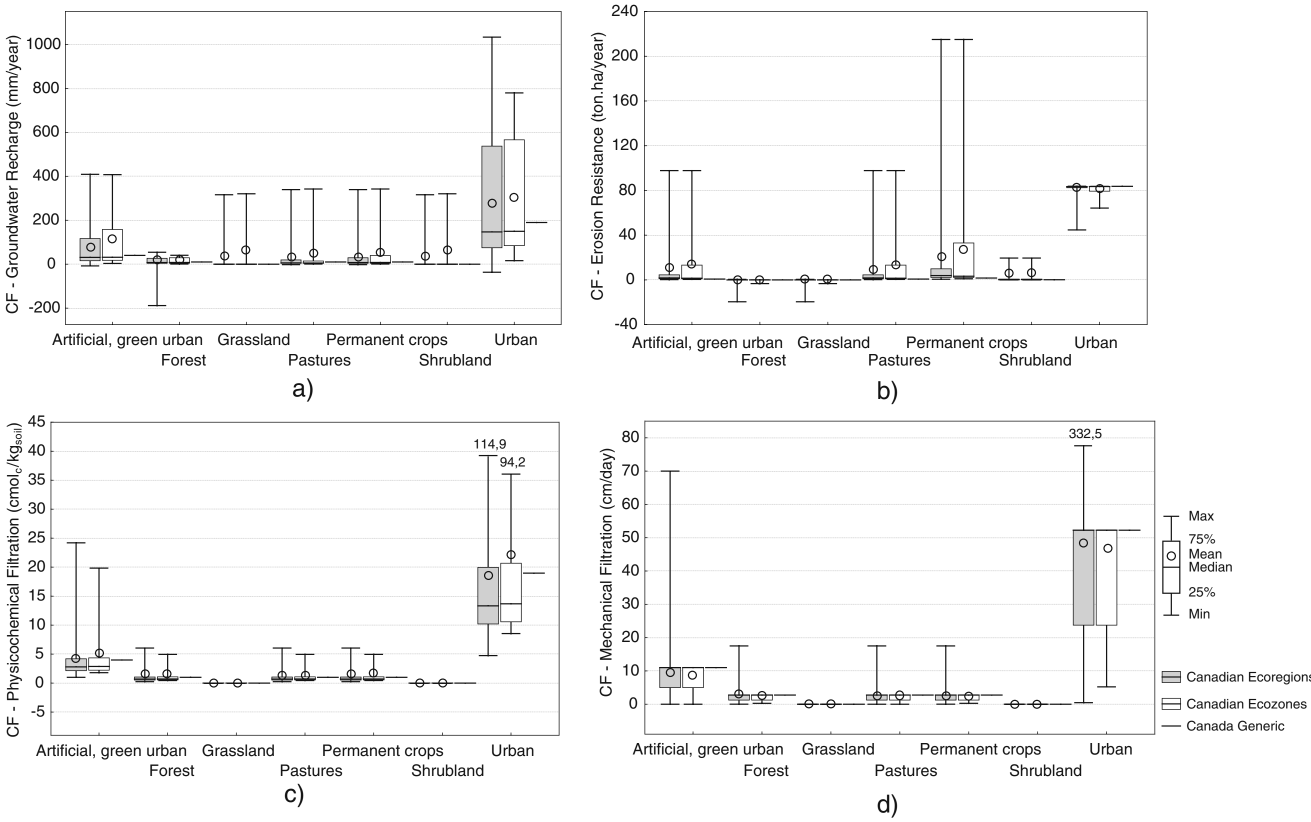

Fig. 6 Comparison of CFs calculated for the three scale levels: the ecoregion (*doted box plot*), the ecozone (*plain box plot*), and the Canadian generic model (*plain line*) and for four impact indicators: **a** groundwater recharge, **b** erosion resistance, **c** physicochemical filtration, and **d** mechanical filtration

the Canadian generic model. Box plot results illustrate the distribution of the CFs of each land use type indicating the extent of spatial variability across and within the ecological units. An important observation that can be made is that besides the similar trend in overall spatial variability noticed for both resolution scales, the one obtained by the ecoregion-based model was higher and revealed interesting variations that were not captured by the ecozone model itself. Depending on the location of the median value for the Canadian generic scale, each of the box plots presents a different skewness pattern. The larger spread of data distribution illustrates the high CF variability for the same use but across several ecological units.

Groundwater recharge capacity When comparing results from all three resolution scales, Fig. 6a shows that generic CFs tend to underestimate at least between 25% (for semi-natural land use types) and 50% (for artificial, agricultural and forestry land use types) of the spatial distribution results. However, no difference in the range sample between both spatial models were observed, except for the urban and the forest land use types where a larger difference was observed for the ecoregion-based scale.

Urban activities can reduce the groundwater recharge capacity for up to 779 and 1,033 mm/year when using respectively the ecozone and the ecoregion-based model for which the associated locations are different. The maximum value from the ecozone resolution scale model refers to the Atlantic Maritime ecozone, whereas the one from the finer scale identifies an ecoregion in the Pacific Maritime, the Queen Charlotte Lowland, as the most potentially impacted region. Moreover, negative CF results were observed when using a finer resolution, suggesting an increase in groundwater recharge capacity in comparison to the corresponding area reference state (PNV). This is likely to occur in arid and semiarid regions characterized by a deficit in hydrological balance where evapotranspiration rates are generally higher than precipitation rates over the year, decreasing the groundwater recharge rate estimate. Located, respectively, in the southeastern and Midwest part of the country, these areas include the Mixedwood Plains ecoregion, showing a CF of up to −188 mm/year, when occupied by forestry use and the Prairie semiarid belt ecoregions, with a CF of up to −36 mm/year, for urban use.

Erosion resistance capacity Erosion resistance capacity could be wrongly estimated for a large number of ecozones and ecoregions when using the generic model. For instance, Fig. 6b indicates that at least 25% of the spatially resolved CFs for urban, forestry as well as grassland tends to be

overestimated while a 50% underestimation can be observed for both green artificial and agricultural use. Moreover, moving to ecozone and ecoregion resolution schemes, a shift can be observed to negative CFs, for forestry and grassland land uses, suggesting an improvement of the erosion resistance capacity. One reason is the influence of the PNV reference state on the results since these major changes were noted in arctic ecozones (Northern and Southern Arctic) and other high altitude areas in the very northeastern part of Canada (Taiga Cordillera), where native vegetation consist of sparse and alpine tundra. Thus, when compared to a densely ground cover such as forest and grassland that improves soil surface stabilization, tundra-poor vegetation is much more vulnerable to erosion ($Q_{\mathrm{use=forest,grassland}} Q_{\mathrm{PNV=tundra}}$).

Moreover, the lowest CF values obtained for the ecozone and ecoregion-based spatial scale are −3.3 and −19 t/ha year, respectively. However, very distinct regions are identified in each case; the coarser resolution highlights the whole Taiga Cordillera ecozone, while the finer one emphasizes the Cascade ranges, an ecoregion in the Pacific Maritime with tundra as the PNV. The higher the resolution, the better the local data (i.e., slope terrain measurement, organic matter content, and vegetation cover) are obtained for and result in a larger spatial variability.

Physicochemical and mechanical filtration capacity As illustrated in Fig. 6c and d, differences in overall spatial variability between ecoregion and ecozone resolution are noted for almost all land use except for semi-natural land use types, grassland and shrubland, for which CFs are equal to zero and no major reduction in filtration capacities are associated to such activities in comparison to the PNV reference state.

The Canadian generic model result falls within the same variability range as that obtained for the ecozone and ecoregion scales. It also underestimates by up to 25% CF results obtained from both spatial models for physicochemical filtration capacity while results for mechanical filtration are overestimated by up to 50%.

Urban land use was shown to be the most impacting land use type for both filtration capacities. More specifically, results for physicochemical filtration reduction ranged from 4 $\mathrm{cmol_c/kg_{soil}}$, in poor northern to southern arctic ecoregions with low CEC, to 114 $\mathrm{cmol_c/kg_{soil}}$ in a northwestern ecoregion located in the Taiga Plains. Results for mechanical filtration illustrate greater differences. The highest reduction in capacity (332 cm/day) occured in ecoregions from the Mixedwood Plains, characterized by sandy soil texture allowing high water permeability. In contrast, for clayey soil texture, such as Taiga Plains ecoregions, a drastically smaller reduction (0.4 cm/day) could be noted.

3.2 Multi-way multidimensional analysis of variance and limitations

Table 4 shows results obtained by the multi-way MANOVA indicating that several input parameters, used in the LANCA® calculation tool model algorithm, significantly influenced ($p<0.05$) the output results (soil ecological quality computation for each impact indicator) while others showed a smaller influence ($p>0.05$) or did not affected them at all ($p=1$). This can be used as guidance with regards to the level of confidence input data parameters have to be collected in.

Based on the nature of the approach, the computation of land quality is significantly affected by the sealing type factor defining each land use type. This may justify the differences or similar behavior observed among the set of land use types tested. Despite the apparent distinctions observed across land use types for the groundwater recharge CFs in Fig. 6a, results from the multi-way MANOVA indicated a smaller influence from the sealing

Table 4 Multi-way MANOVA results on soil ecological functions parameter computed by the LANCA® calculation tool model

Input parameter	Groundwater recharge	Erosion resistance	Physicochemical filtration	Mechanical filtration
Humus content	0.927	0.941	1.000	1.000
Gravel content	0.767	0.000*	1.000	1.000
Precipitation	0.000*	0.957	1.000	1.000
Summer precipitation	0.715	0.292	1.000	1.000
Soil texture	0.785	0.000*	1.000	0.000*
Sealing type	0.739	0.000*	0.000*	0.000*
Evapotranspiration	0.000*	0.906	1.000	1.000
Slope	0.718	0.000*	1.000	1.000
Depth to groundwater	0.904	0.370	1.000	0.000*
CEC	0.577	0.986	0.000*	1.000

*$p<0.05$

type (p=0.739). This can be explained by the fact that precipitation and evapotranspiration rates have a significant influence and are extremely dominant in the model, overwhelming the output results.

Based on a typical water balance, natural groundwater recharge is the difference between precipitation, evapotranspiration and runoff rates (Lerner 2002). Urbanization effects disrupt water balance often by reducing infiltration rates to groundwater and increasing water runoff due to presence of paved areas. Hence, natural groundwater recharge from precipitation is diminished. However, mains water leakage is identified in the literature as an additional greater indirect source for groundwater recharge in urban areas. But due to the complexity of city infrastructures, it is often neglected (Foster et al. 1993; Lerner 2002). Interestingly, although negative CFs for urban use were obtained in Canadian semiarid areas (Prairies) suggesting an improvement for groundwater recharge capacity, such results can be inconsistent and seem to be slightly overestimated. The reason is that the calculation estimated for groundwater recharge capacity in the LANCA® model only considers traditional natural recharge from precipitation and is rather based on a runoff coefficient, failing to account for a runoff flow and resulting in a negative balance.

Erosion resistance capacity is driven by several parameters such as gravel content, soil texture, slope (steepness of land cover) and sealing type. Indeed, as observed in all Canadian Cordillera ecozones, a finer texture with a weak structure and high steepness enhances soil loss and consequently decreases its erosion resistance capacity resulting in high CFs for several land use types. Conversely, coarser soil texture and the presence of ground vegetation cover help reduce soil loss. These cases can be observed in the Boreal and Taiga Plains which have coarse and silty loam soil texture. In addition, although it is a hilly region, occupying the Taiga Cordillera ecozone by a dense groundcover such as grassland or forest can become beneficial compared to the tundra natural vegetation state. As mentioned earlier, the same conclusion can be drawn for arctic ecosystems.

In addition to the sealing type, both filtration capacities are influenced by the soil vulnerability and its inherent and highly site-dependant attributes. On one hand, two major parameters affect the mechanical filtration capacity, soil texture and depth to groundwater. On the other hand, CEC influences the physicochemical filtration capacity. As a result, calculating values for the input parameter aggregated on a mean basis and normalized for each ecological unit can justify the small spatial variability across ecozones and similarities in CF results (see Online Resource 2). The estimates for soil filtration capacities can be averaged out when different zoning systems are used. For instance, at a larger scale, many Canadian soil types have a similar texture (loamy and sandy loam) while the finer ecoregion-based scale allows considering different ones. Thus, the LANCA® model can seem inappropriate when used for calculating a default CF at a coarse scale level. Moreover, in regards to the sealing type, no distinction is made between agricultural land use activities and forestry for both filtration capacities (see Fig. 6c and d). This can be a limitation from the LANCA® model since vegetation type and root biomass uptake may have a different influence on infiltration rates, controlling water storage and consequently mechanical filtration capacity.

Based on the observations and result discussion, the model was found to be generally too coarse and can require significant input increments to change the quality result of an impact indicator. This can be one reason why only extreme input value changes (very high or low values) were observed when using an ecoregion-based model. Furthermore, a comparison between all pairs of CF results did not reveal any correlation between the four impact indicators, indicating that they are independent.

4 Conclusions and recommendations

This work demonstrates the feasibility of developing a characterization model and factors to address impacts on soil ecological functions, for seven different land use types and taking into account spatial differentiation. Impacts of land transformation are proportional to the area of land transformed and the difference in soil ecological quality (ΔQ) multiplied by the relaxation time needed for the ecosystem to reach the PNV state (Eq. 3). The equivalent impact between occupation and transformation is reached when $t_{\mathrm{occ}} = \frac{1}{2} \times t_{\mathrm{relax}}$. The first parameter is given by the LCI and the second one depends on the dynamic of the vegetation cover in a given bio-geographical condition. Despite the difficulty in measuring such dynamic parameters, many publications (Müller-Wenk 1998; Weidema and Lindeijer 2001; Koellner and Scholz 2007; Schmidt 2008) already propose a non-exhaustive list of relaxation time estimates that can be used for developing land transformation CFs.

CFs showed spatial variability up to a factor of 8 across Canadian regions depending on the location of use. Such spatial variability is strongly related to inherent natural variability and spatial heterogeneity across several types of ecosystems captured by the collected input parameters and for which the chosen geographical boundaries are deemed adequate. This variability gives a first estimate of the uncertainty linked to the Canadian generic CF when spatial differentiation is not taken into account. A higher resolution, such as the Canadian ecoregion-based scale is found to be more appropriate, because it can bring additional

discrimination. However, the efforts required for collecting more spatial-specific data could be offset by the coarse discrimination capacity of the LANCA® calculation tool model.

Highlighting the relevance of addressing spatial differentiation for land use, the operationalization of this method at a global scale could be performed using a similar spatial approach based on a worldwide biogeographical boundaries such as the Holdridge Life Zones classification (Holdridge 1947); defining nine broad bioclimatic regions and 38 subregions. Such ecological boundaries are considered appropriate as they take into account a range of geospatial parameters.

A practical implementation of this method in LCA requires an identification of the ecosystem type where the activity is taking place, in addition to defining the land use type, the area size during the activity (square meters) and the time required for the occupation (year). Alternatively, since the method depends on several input parameters that can require different spatial scale levels of detail, a differenciation between situations, as suggested by Milà i Canals et al. (2007a), can be performed by considering a restricted set of CFs for archetypal scenarios. This may be carried out based on the multi-way MANOVA results identifying the most influenctial parameters that should be collected with high confidence and defining default values for specific scenarios (for instance, urban use in rich soil and high precipitation rate). Therefore, by choosing a range of priority parameters, different types of spatial scales can be defined for each soil ecological function. Hence, mechanical filtration and groundwater recharge capacities would be better suited, respectively, with a soil texture and bioclimatic region delineation differentiating between wet and dry regions.

Some recommendations can already be made using the LANCA® model, which is an appropriate tool from an LCA perspective, yet requires some improvements to sophistication in the algorithm to better refine differentiation and stratification between several intensive land use processes. It would also be interesting to integrate an additional factor, correcting the impact on each ecological function capacity that takes into account the regional scarcity as established in the Swiss Ecological Scarcity method (Frischknecht et al. 2008).

The choice of several indicators addressing land use impacts on a range of soil's ecological function capacities can vary depending on the scope of the study and be further discussed for selecting the most appropriate ones and limiting their quantity. An uncertainty assessment should be performed as well, in order to better understand the influence of the model and the parameters on the final results. It would also be possible to evaluate the results obtained when using the LANCA® calculation tool model by comparing them with simulation results using more sophisticated models, such as: Erosion Productivity Impact Calculation, Chemicals, Runoff, and Erosion from Agricultural Management Systems, and Water Erosion Prediction Project. Alternatively, they may be a useful starting point for simulating and predicting soil ecological quality change associated to land use (Arshad and Martin 2002).

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